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Ecosystem services: Urban parks under a magnifying glass?

Teresa Mexiaa,1 , Joana Vieiraa , Adriana Príncipea , Andreia Anjosa , Patrícia Silvab , Nuno Lopesb , Catarina Freitasb , Margarida Santos-Reisa , Otília Correiaa , Cristina Branquinhoa,? , Pedro Pinhoa,c a Centre for Ecology, Evolution and Environmental Changes (cE3c), Faculdade de Ciências, Universidade de Lisboa, 1749-016 Lisboa, Portugal b Divisão de Estudos e Gestão Ambiental e de Energia, Departamento de Energia, Clima, Ambiente e Mobilidade, Direção Municipal de Ambiente, Mobilidade, Energia e Valorização Urbana, Câmara Municipal de Almada, Casa Municipal do Ambiente, Rua Bernardo Francisco da Costa, 42, 2800-029 Almada, Portugal c Centro de Recursos Naturais e Ambiente, Instituto Superior Técnico, Universidade de Lisboa, Av. Rovisco Pais, 1049-001 Lisboa, Portugal

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ABSTRACT Urban areas' population has grown during the last century and it is expected that over 60% of the world population will live in cities by 2050. Urban parks provide several ecosystem services that are valuable to the wellbeing of city-dwellers and they are also considered a nature-based solution to tackle multiple environmental problems in cities. However, the type and amount of ecosystem services provided will vary with each park vegetation type, even within same the park. Our main goal was to quantify the trade-offs in ecosystem services associated to different vegetation types, using a spatially detailed approach. Rather than relying solely on general vegetation typologies, we took a more ecologically oriented approach, by explicitly considering different units of vegetation structure and composition. This was demonstrated in a large park (44 ha) located in the city of Almada (Lisbon metropolitan area, Portugal), where six vegetation units were mapped in detail and six ecosystem services were evaluated: carbon sequestration, seed dispersal, erosion prevention, water purification, air purification and habitat quality. The results showed that, when looking at the park in detail, some ecosystem services varied greatly with vegetation type. Carbon sequestration was positively influenced by tree density, independently of species composition. Seed dispersal potential was higher in lawns, and mixed forest provided the highest amount of habitat quality. Air purification service was slightly higher in mixed forest, but was high in all vegetation types, probably due to low background pollution, and both water purification and erosion prevention were high in all vegetation types. Knowing the type, location, and amount of ecosystem services provided by each vegetation type can help to improve management options based on ecosystem services trade-offs and looking for win-win situations. The trade-offs are, for example, very clear for carbon: tree planting will boost carbon sequestration regardless of species, but may not be enough to increase habitat quality. Moreover, it may also negatively influence seed dispersal service. Informed practitioners can use this ecological knowledge to promote the role of urban parks as a nature-based solution to provide multiple ecosystem services, and ultimately improve the design and management of the green infrastructure. This will also improve the science of Ecosystem Services, acknowledging that the type of vegetation matters for the provision of ecosystem services and trade-offs analysis.

1. Introduction

Over the last century, the population of urban areas has grown considerably. By 1950, these areas accounted for 30% of the world's population, nowadays reaching 50%, and it is estimated to grow above 65% by 2050 (United Nations, 2014). Urban areas only cover about 2.4% of the land area (Millennium Ecosystem Assessment, 2005a), thus having a very high population density. In this way, ecosystems in urban

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? Corresponding author. 1 Present address: Centro de Estudos Florestais, Instituto Superior de Agronomia, Universidade de Lisboa, Edifício Prof. Azevedo Gomes, Tapada da Ajuda, 1349-017 Lisboa, Portugal. E-mail addresses: teresa.mexia@gmail.com (T. Mexia), jivieira@fc.ul.pt (J. Vieira), aprsilva@fc.ul.pt (A. Príncipe), asanjos@fc.ul.pt (A. Anjos), pasilva@cma.m-almada.pt (P. Silva), nlopes@cma.m-almada.pt (N. Lopes), cfreitas@cma.m-almada.pt (C. Freitas), mmreis@fc.ul.pt (M. Santos-Reis), odgato@fc.ul.pt (O. Correia), cmbranquinho@fc.ul.pt (C. Branquinho), ppinho@fc.ul.pt (P. Pinho).

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T areas play an increasingly key role in the well-being of the inhabitants of these highly modified landscapes (Chiesura, 2004; Gómez-Baggethun and Barton, 2013; Pinho et al., 2016). Although their ecological value has often been considered limited due to their size and degree of artificiality (Davies et al., 2011), urban ecosystems can provide various ecosystem services (hereafter referred to as ES), i.e. benefits that man receives from ecosystems (Millennium Ecosystem Assessment, 2005; TEEB, 2011). These ES are the base for the use of these ecosystems as a nature-based solution to multiple environmental problems that are frequent in cities. Among urban ecosystems, parks provide several services, such as water and air purification, wind and noise reduction, carbon sequestration, microclimate regulation, wildlife habitat, and social and psychological well-being (Chiesura, 2004; Millennium Ecosystem Assessment, 2005). However, in natural ecosystems, the type and magnitude of the ES provided depend on their characteristics, such as vegetation type, so we should expect the same to occur in urban ecosystems. For example, different management practices may implicate a trade-off between which service is maximized (different tree planting schemes for pollution removal vs. heat mitigation; Bodnaruk et al., 2017), or between ES and ecosystem disservices (increasing carbon sequestration by vegetation vs. carbon emissions due to its maintenance (e.g. mowing, pruning, irrigation, and fertilization); Jo and McPherson, 1995). Because trade-offs occur with management options, the assessment of multiple ES could inform decision-makers and provide planning options that may enhance the value of urban parks as nature-based solutions in the provision of ES, and thus improve quality of life in urban areas (Haase et al., 2014). There are several studies that evaluate ES in different typologies of urban green areas (street trees, parks, private gardens, etc.; e.g. Derkzen et al. (2015), Nowak (1993), Strohbach and Haase (2012), Sutton and Anderson (2016)) and in areas with the same typology but different types of management (Lilly et al., 2015; Qian et al., 2010). However, no study was found comparing the effect of different vegetation types in multiple ES within the same park. Among those studying one or several parks in detail, i.e. considering different management or land-cover types, only a single service was studied. For example, Bae and Ryu (2015) and Gratani et al. (2016)) studied the effects on carbon sequestration, while Speak et al. (2015) focused on biodiversity. Derkzen et al. (2015) went one step further and studied multiple ES provided by the entire green infrastructure. These authors separated green infrastructure according to type (e.g. isolated trees versus park) and could provide high-resolution maps of bundles of ES. Here we propose to consider a more ecologically oriented view, by explicitly looking at the effect of vegetation structure and composition, as well as local orography, on ES. This approach was tested in a single large urban park. Large parks can be highly heterogeneous regarding vegetation types and, in addition, they may also be subject to multiple management options. As such, it is possible to make use of a spatially detailed mapping of vegetation types and orography to study the associated ES. In this work, we mapped and quantified six ES – carbon sequestration, seed dispersal, erosion prevention, water purification, air purification and habitat quality – provided by different vegetation types that are common in urban parks. Considering that urban green spaces may have an important role in carbon sequestration at a local level (Niemelä et al., 2010), carbon sequestration was assessed through carbon stock estimation from vegetation biomass, litter, and soil measurements. Aside from the direct effect of vegetation on carbon balance, and even if the sequestered volume is lower than the CO2 produced by the surrounding city (Lebel et al., 2007), the increase of green areas affects microclimate and reduces the heat island effect on cities, promoting an indirect reduction of CO2 emissions (Hardin and Jensen, 2007; Niemelä et al., 2010; Nowak, 1993). Abundance of omnivorous birds was assessed as a surrogate of seed dispersal. This is a service that is provided by animals, as well as pollination, and it is crucial for the reproduction of plants (Herrera and Pellmyr, 2002), allowing their establishment and self-sustainability. Erosion prevention and water purification were assessed using InVEST - Integrated Valuation of Ecosystem Services and Tradeoffs v.3.2.0 (Natural Capital Project, 2015), an open-source software that maps and valuates ES (Sharp et al., 2016). Vegetation and soil can filter urban effluents, reducing pollutants and nutrient levels (TEEB, 2011), a feature that is essential for the maintenance of groundwater quality, as it is frequently used for irrigation or human consumption. This is particularly important, since urban water drainage systems often have high concentrations of nutrients (Nidzgorski and Hobbie, 2016) that are malodorous, may increase water turbidity, and can cause water and soil eutrophication, thus compromising their quality (Millennium Ecosystem Assessment, 2005). Moreover, vegetation also prevents erosion, by reducing runoff, retaining sediments, and stabilizing soil, preventing landslides and floods (Gómez-Baggethun and Barton, 2013; López-Vicente et al., 2013; Verstraeten et al., 2006). The air purification service was evaluated through the assessment of epiphytic lichens' species richness. This lichen metric has been used as a general surrogate of atmospheric air pollution, because the total number of lichen species decreases with intense pollution (Kapusta et al., 2004; Langmann et al., 2014; Llop et al., 2012; Svoboda et al., 2010). Air pollution is a common problem of urban environments, and air quality may be promoted by the presence of vegetation, with trees having a positive effect on it by filtering atmospheric particulates (Bolund and Hunhammar, 1999; Derkzen et al., 2015). Finally, habitat quality was evaluated using oligotrophic epiphytic lichens' abundance, not only because it is connected to overall habitat conservation conditions, but also because it is well related to other biological groups of interest to conservation such as birds (Llop et al., 2012; Pinho et al., 2016). These lichens' functional group is extremely sensitive to many disturbances, including the many problems associated with increasing urbanization – decreasing with eutrophication and dust pollution –, and it is also sensitive to habitat stability and management intensity (Llop et al., 2012; Munzi et al., 2014; Pinho et al., 2011). Our main goal was to quantify trade-offs and synergies in ES that are highly relevant to the sustainability of urban ecosystems and to human well-being, associated with vegetation type and management. This was done through a novel approach, by looking at vegetation structure and composition, and using spatially detailed vegetation cartography and orography mapping. We hypothesized that ES provision would vary with a park's vegetation type. This will improve the knowledge on urban ecosystems and provide local authorities with tools to optimize ES supply through management and local planning. 2. Material and methods 2.1. Study area The study area was an urban park ("Parque da Paz") with approx. 44 ha located in the city of Almada, Portugal (38°9.771'N, 9°9.828'W, Fig. 1). Almada is one of the most populated municipalities of Portugal, with an area of approx. 70 km2 and 174 030 habitants (INE, 2012), located within the Lisbon metropolitan area (south-west Europe), under Mediterranean climate. The park was established in 1997, in lands used formerly for agricultural purposes that had once been assigned to urbanization, and it is currently the city's largest urban green. It is surrounded by roads, including highways with high traffic intensity. The park's topography was modeled by construction to regulate its hydrology, to diminish air, visual, and noise pollution (Pardal, 1997), and its altitude ranges from 7 m to 48 m high. 2.2. Land-cover and vegetation units Land-cover characterization of the park (see Fig. 1) was based on the general municipality cartography of the park. A more detailed classification of the vegetation units was then made through the T. Mexia et al. Environmental Research 160 (2018) 469–478 470 analysis of aerial and satellite imagery available online (resolution 0.5 m, Digital Globe (ArcGIS Online basemap, World Imagery)), in order to identify different vegetation types. Afterwards, field visits were done to validate the obtained cartography. 2.3. Carbon sequestration Carbon stock was used as a surrogate of carbon sequestration. Carbon stock of each vegetation unit was calculated as the sum of trees, shrubs, herbs, litter, and soil carbon pools, estimated from field measurements made in September 2015 and using literature values. Above and below-ground woody species' dry-weight biomass was calculated using allometric equations from the literature (see Supplementary Table S2). To do so, height and diameter at breast height (or crown diameters for shrubs) were measured in a minimum of 5 individuals of tree and shrub species for the whole study area. When species size varied significantly (e.g. recently planted areas and older areas), size classes were defined and at least 5 individuals were measured for each class. Species-specific equations were selected from studies done in the Mediterranean basin, whenever possible. When no species-specific parameters were available, parameters calculated for the same genus or other taxonomical level were applied (broadleaf, conifer, etc.). Individuals whose measures did not comply with the equations' requirements (e.g. DBH range), or species with fewer than 5 individuals, were excluded from the analysis. Plant density (trees and shrubs) was determined through direct counting, except in the mixed forest area, where a sampling was carried out with 6 quadrats of 10mx10m (since this is a homogeneous patch of remnant vegetation). Quadrats were randomly placed using the Random points tool from ArcGIS, constrained by a 10 m buffer from margins and main trails. All tree and shrub individuals within the sampling squares were measured. Species' average biomass was then multiplied by the number of individuals present in each vegetation unit. In the mixed forest, biomass was calculated for the sampled area and then extrapolated to the total area. Carbon content of woody species was obtained through the application of a conversion factor of 0.50 (IPCC, 2003) to the above and below-ground biomass. Non-woody species' carbon content was estimated for lawns and grasslands, where biomass was considered relevant for total carbon stock. Sampling was made at three random points within both vegetation units, by harvesting the above-ground biomass of four 20 × 20 cm (0.04 m2) quadrats in each point, clipped as close to the ground as possible. Samples were weighed after drying in the oven at 60 °C for 72 h. Samples were milled until they were a fine powder and the amount of carbon was determined through elemental analysis using the Dumas method of combustion. The percentage of carbon was then multiplied by the sample dry weight, to estimate the total stock of carbon in each area. Herbaceous below-ground carbon content was estimated using values from the literature (see Supplementary Table S3). For the estimation of soil carbon content, we took three samples, composed of 10 randomly distributed subsamples, for each vegetation unit. Soil was sampled up to 10 cm depth using 2 cm diameter cores. Samples were oven-dried for 72 h at 60 °C, milled and the amount of carbon was determined through elemental analysis using the Dumas method of combustion. The carbon stock was calculated using the following equation: $CT = CF \times D \times V$, where CT is total carbon for the sampled layer of soil (Ton), CF is the fraction of carbon, D is density (kg/m3), and V is volume of the soil layer (m3 ; V = Area x Depth) (Donovan, 2013). Finally, litter carbon content was estimated from literature values (see Supplementary Table S3). Carbon content of the vegetable garden was not assessed in the field due to its high heterogeneity in cultivation and land maintenance, besides logistic constraints. This vegetation unit was considered less relevant, since it is not currently managed by the park's administration. Nevertheless, its tree and shrub cover is sparse, and herb and litter cover is frequently removed during harvest, so we estimated the soil carbon pool for this area using literature values (Supplementary Table S3).

Fig. 1. General location of the study area (red squares and cross) in Europe and the Lisbon metropolitan area, showing also the surrounding land-cover (based in the European Urban Atlas, 2012), as well as the park's land-cover and mapping of vegetation units, based on vegetation types. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

2.4. Seed dispersal

The abundance of omnivorous bird species was used as a surrogate of the seed dispersal service. Bird counts of all species were held in June 2015, between 8 and 10 a.m. and in suitable weather conditions. For each type of vegetation unit, one point was selected and sampling was always performed by the same observer. At each point, the number of individuals and species present within a radius of 50 m from the observer was recorded for periods of 10 min, through their acoustic and visual detection (Bibby and Burgess, 2000; Sutherland et al., 2004). Birds recorded as only flying over the area were omitted from analysis, since it was not clear if they interacted with the habitat or not (Jasmani et al., 2017; Strohbach et al., 2013). Species richness and abundance of birds were assessed for each point, and species were sorted by diet, based on the works of Herrera (1995), Lizée et al. (2011), Santana et al. (2012) and Svensson et al. (2009). Omnivorous species that are considered preferential frugivorous were considered in a separate group as main seed dispersers. Birds were not assessed in the park's avenues and parking lots, due to their small area, and in the vegetable garden.

2.5. Erosion prevention and water purification

Water purification and erosion prevention services were assessed using the software InVEST – Integrated Valuation of Ecosystem Services and Tradeoffs (Natural Capital Project, 2015). InVEST is an open-source software that includes several spatially-explicit models to map and value different ES, from a local to a global scale (for a revision on ES modelling software please see Bagstad et al., 2013 and Olosutean, 2015). For erosion and water purification services, InVEST uses two main sets of parameters. The first parameter is a digital terrain model, used by InVEST to calculate slope or amount of draining cells in each cell. This digital terrain model was built using level contours, provided by the municipality, through ArcGIS contour interpolation adjusted for hydrographic correctness (ESRI, 2015). The final digital terrain model had 1 m spatial resolution. The second set of parameters is land-cover data, and associated values of erosion and water purification. For landcover methods and types see 2.2. As for the parameters depicting erosion (sediments) and water purification (nitrogen flows), these were associated to each land-cover type (e.g. the total amount of nitrogen typically provided to lawns). For more details on the parameters used and bibliographic references see Supplementary Table S.1.1. The model Sediment Delivery Ratio was used to assess erosion prevention, by mapping the amount of sediments generated and delivered to streams by each sub-basin (Sharp et al., 2016 for details). The model Nutrient Retention: Water Purification was used to assess water purification by the ecosystem. Nitrogen was used as a surrogate of overall nutrient retention, because it is extremely well related to phosphorus retention. After running both models, the final output (retention and export of sediments and nutrients to the watershed, also with 1 m resolution) was averaged per vegetation unit and retention percentage was calculated in relation to total loads. 2.6. Air purification and habitat quality Lichen diversity was used as a surrogate of both air purification and habitat quality, by using different metrics. Epiphytic lichen diversity was measured in September 2015 in 29 trees of Quercus spp., using the European standard methodology (Asta et al., 2002; Cristofolini et al., 2014). Tree selection was done by stratifying for vegetation unit, ensuring that all units were sampled, and sampling five locations per unit. In each location, tree selection was done ensuring that the tree was within the protocol requirements (trunk's sampling portion at more than 100 cm from the ground and with a minimum circumference of 50 cm, small inclination, no visible injuries or disease). Furthermore, an effort was done to sample trees that were well distributed throughout the park and under homogeneous conditions for each vegetation unit. However, due to local conditions (plantation scheme), this was not entirely possible and some clustering occurred, but sampling was still well distributed over the entire park. In selected trees, lichens were sampled using a grid with five 10 × 10 cm squares. This grid was placed on the trunk at a minimum height of 1 m from the ground and at the four cardinal points. Species frequency was measured as the number of squares in which each species was found. From this data, the total number of species was used as a surrogate of air purification. For habitat quality, species were classified into functional groups, based on their maximum tolerance to eutrophication (Nimis and Martellos, 2008). The sum of the frequency of all oligotrophic species (LDVoligo, sum of species with classification 1 and 2 provided by P. Nimis and Martellos, 2008) was used as a surrogate of habitat quality. Tree diameter at breast height was measured to test if it influenced lichen metrics, but no significant relations were found (data not shown, $p < 0.05$). Variogram analysis revealed a good spatial structure for both metrics, with a nugget effect of zero and a spatial continuity of 500 m for both. A spherical model without anisotropy was fitted to this experiment variogram and was used to interpolate both lichen metrics within the park using ordinary kriging, and then the average values were determined within each vegetation type. 2.7. Data analysis Firstly, all absolute values measured or estimated are reported per vegetation type. Whenever values were obtained from 1 m resolution outputs, the standard deviation is provided per vegetation type. Secondly, we compared the provision of multiple ES by each vegetation type. For that, we standardized the results obtained for each service in relation to the maximum value observed in the park for that service and obtained a score for each vegetation type, varying from 0 to 1, where 1 corresponds to the highest value. The average of this score per vegetation type (average of all ES score, assuming equal importance for all services) was provided as an integrative measure of all tested ES and as a framework to evaluate trade-offs between them. 3. Results 3.1. Land-cover and vegetation units A total of eleven land-cover classes were identified in the park area, including six vegetation units covering c. 77% of the total area (Fig. 1 and Table 1; for a more detailed description see Supplementary Table 4). Each vegetation unit corresponded to a main vegetation type, which was classified based on tree density (low and high tree density) and type of understory (no understory – trails and parking areas; irrigated herbaceous understory – lawns; not irrigated herbaceous understory – grassland). Alongside planted areas, there is a well-preserved area of Mediterranean forest and an area of small patches for agricultural use. The forest area had the highest tree cover of the park and was the only area with a well-developed shrubby understory and no significant herbaceous cover. 3.2. Comparison of ecosystem services supply by the different vegetation types The park's total carbon stock was 4040Mg, with a greater contribution from areas with the highest tree density (forest and grassland with high tree density). Although these areas represent close to 30% of the park's areas with plant cover, they accounted for 70% of the total carbon stock. Grasslands with high tree density areas and the forest had 228 and 262 Mg C ha-1 , respectively (Fig. 2 – a). As expected, when tree density is lower, the carbon stored in non-woody vegetation and in soil is more relevant for the total carbon stock, although it was only estimated for the first 10 cm of soil. Finally, the amount of carbon stored in trees in park avenues and parking lots (classified as urban tree T. Mexia et al. Environmental Research 160 (2018) 469–478 472 overstory) was similar to that estimated for trees in areas with lower planting density. Total bird abundance was higher in both areas with lower tree density (Fig. 2 – b). Potential seed dispersal service, indicated by the abundance of all omnivorous birds, was higher in lawns, but main seed dispersers were found in all sampled vegetation types. From the 11 omnivorous species identified in the study area (in a total of 18), 4 are important seed dispersers in the Mediterranean region. Erosion prevention and water purification services' supply was homogeneous throughout the park, with nutrient and sediment retention approaching 100% for all vegetation types (Fig. 2 – c, d). Although greater nutrient inputs were associated with lawns and the vegetable garden, retention was also higher in these areas, resulting in overall leaching approaching zero. As expected, potential sediments production was higher in areas with greater slope, but their loss was prevented by high retention efficacy of these areas, which are also more heavily forested (forest and grassland with high tree density). In relation to air purification (Fig. 2 – d), this service also had similar values for all vegetation types. The average number of lichen species was higher in the mixed forest (c. 12 species), while other vegetation types presented values of approx. 9 species. Regarding habitat quality (Fig. 2 – e), the best quality was observed in the forest (LDV of oligotrophic lichens of approx. 20). This vegetation type presented c. the double of any other area, and these results were independent of tree density or understory management in planted areas. 3.3. Integrated ecosystem services' assessment When all services were standardized and integrated, it was evident that the mixed forest area had the maximum total supply of all tested ES (Fig. 3). Overall, the services that showed less dependence from vegetation type were related to water purification, but also to air purification, while carbon stock was clearly dependent on tree density, and habitat quality was only optimized in the mixed forest area. As a consequence of reaching the highest possible provision of most ES, the average score of the mixed forest was 0.94. The grassland with low tree density presented the lowest average score (0.60), while the more heavily planted area reached 0.71. 4. Discussion The spatially detailed analysis of the provision of ES by an urban park has revealed that it may not function as a homogeneous entity. Rather, it was shown to have multiple facets that depend on the type of vegetation, its management, and local topography. On the one hand, some of the ES selected varied greatly with vegetation type, namely carbon sequestration, seed dispersal, and habitat quality. On the other hand, erosion prevention, and water and air purification services had similar values in all vegetation types. By using this information, practitioners can optimize the management of urban parks as a nature-based solution to either promote a given service or to balance several ES of interest. This was only possible by simultaneously assessing multiple ES and by performing a spatially detailed characterization of local topography and vegetation distribution throughout the park. 4.1. Ecosystem services supply Overall, the vegetation type that provided the higher integrated score for ES was the mixed forest. In fact, the forest presented the maximum value for almost all tested services. Its advantage over the other vegetation type with a close overall value (grasslands with high tree density) was mainly related to habitat quality, measured with oligotrophic lichen species, which are very sensitive to most types of disturbances, including the impact of urbanization (Pinho et al., 2016, 2011). This result was expected, as the mixed forest was chosen by park managers to preserve local biodiversity, and it is the area that most resembles a natural vegetation structure within the park. This area is much older than the rest of the park: it is only occasionally subjected to intervention and it is less used by visitors. Its vegetation structure is more complex, with a well-developed shrubby understory and a dense tree cover, creating refuge conditions for fauna, by providing shelter and food, and increasing its diversity, which is especially important in urbanized areas (Barth et al., 2015; Heckmann et al., 2008). However, it is worth noting that planted areas with lower tree density showed higher overall abundance of birds than the mixed forest. This effect may be explained by a higher species diversity and a more heterogeneous distribution of planted species that create a multiplicity of niche and can increase the biodiversity of such planted areas (Bolund and Hunhammar, 1999). Carbon sequestration was the service more obviously related to vegetation type, being clearly higher in vegetation types with the highest tree density. The carbon stock in these areas (forest: 262.35Mg ha-1 ; grassland with high tree density: 228.10 Mg ha-1 for) was higher than the one estimated for European forests by FAO (2010), with an average of 161.8 Mg ha-1 , which is probably related to the high tree density in our study area. In fact, the estimated carbon for trees' aboveground biomass (forest: 166.95Mg ha-1 ; grassland with high tree density: 156.30Mg ha-1) had values close to those observed in pine forests in Portugal (Correia et al., 2010; maximum value of approx. 200Mg ha-1), which are considered high-density plantations. Although in the remaining areas the estimated carbon stock for trees (above and below-ground) approached the values calculated for other urban green areas (Dorney et al., 1984: 18.30 Mg C ha-1 ; Rowntree and Nowak, 1991: 27 Mg C ha-1), the comparison is not always possible or easy to perform, since the estimated values depend on many factors, ranging from the characteristics of the green space itself (e.g. planting density, tree age) to the method used for the estimation. In other works, for example, Strohbach and Haase (2012) estimated 29.39 Mg C ha-1 for the aerial biomass of trees in the city of Leipzig, while Nowak (1993) estimated 8.8 Mg C ha-1 for green spaces within residential areas. In our study area, the carbon stock of lawns was lower than in the other vegetation types; however, lawns are very common in urban parks, and they represent an important fraction of urban ecosystems (Lilly et al., 2015), so their carbon stock may be very relevant carbon balance in urban areas. In our case, estimated values for lawns' grass and soil Table 1 Land-cover classes defined for the study area, total area (ha) and relative area (%), in relation to the total study area and to the total area with plant cover, and tree density (n° ind. ha-1), when applicable (n.d. – no data). Land-cover classes Area (ha) Relative area (%) Tree density (n° ind. ha-1) Total Vegetation Vegetation units (vegetation types) 38.4 86.6 Grassland with high tree density 6.0 13.5 15.6 382 Grassland with low tree density 11.9 26.8 30.9 156 Lawn with low tree density 9.5 21.5 24.9 99 Urban tree overstorey 1.2 2.7 3.1 183 Vegetable garden 4.2 9.4

10.9 n.d. Mixed forest 5.6 12.7 14.6 467 Other units 6.0 13.4 Trails 3.2 7.2 Urban infrastructures 1.3 3.0 Lake 0.8 1.8 Stream 0.2 0.5 Wasteland 0.5 0.9 Total area 44.4 T. Mexia et al. Environmental Research 160 (2018) 469–478 473 carbon were much higher than those estimated for other similar areas (grass: 1.86 Mg ha-1 (Jo and McPherson, 1995) and grass and soil (< 10 cm): 6.04 Mg ha-1 (Lilly et al., 2015)). Once more, their management influences the stored carbon. For example, part of the biomass produced in non-irrigated grasslands is incorporated into soil, increasing carbon sequestration in these areas (Machmuller et al., 2015), an effect that greatly increases in fertilized or irrigated areas (Conant et al., 2001). Also, paved areas with tree overstory often take up a significant area in urban parks and their surroundings, so their contribution to the total carbon stock may also be important, and the inclusion of trees should be taken into account when designing and managing these areas (O'Donoghue and Shackleton, 2013). Regarding the seed dispersal service, we considered all omnivorous bird species to be potential seed dispersers, since their diet may include feeding on fruit at some time. Nevertheless, some preferential frugivorous species have been pointed out as the main seed dispersing birds in the Mediterranean. These species belong to the genus Sylvia, Turdus and Erithacus (Herrera, 1995) and were all represented in the park, often with higher abundance than other omnivorous. A higher abundance of frugivorous birds increases the number of seeds that can be dispersed and, in turn, a greater species richness increases the variety of different types and sizes of fruits that can be dispersed (Herrera, 1995; Jordano, 2000). As such, omnivorous birds may be a better surrogate of the ability to disperse seeds than overall species richness. In fact, we observed a similar total species richness among vegetation types, contrary to what was observed by Shwartz et al. (2008) in a 262 ha urban park, which may be related to our park's size. We did not assess bird mobility nor visiting intensity, but most of our study area has only moderate use, so we believe that trails did not reduce bird crossing between patches (Fernández-Juricic and Tellería, 2000) and that bird presence was determined by the mosaic of different habitats within the park. However, since birds were sampled at feeding time, functional groups' abundance in each vegetation type may be more related with food availability (Moorcroft et al., 2002; Ward and Zahavi, 1973). Consequently, it may determine seed dispersers' presence, thus being a good indicator of this ES potential provision. Urban ecosystems harbor Fig. 2. Ecosystem services evaluated for each vegetation type: a – carbon sequestration (carbon stock; Mg/ha), b – seed dispersal (bird abundance per trophic group), c – erosion prevention (sediment retention; %), d – water purification (nitrogen retention; %), e – air purification (lichen species richness; number of species), and f – habitat quality (LDV with only oligotrophic lichen species; species frequency); mean ± SD were calculated for services estimated per area unit (1 m2 resolution). T. Mexia et al. Environmental Research 160 (2018) 469–478 474 important bird populations (Melles and Martin, 2003), whose functional diversity can in this way be enhanced by appropriate management measures, such as the number of habitats in the park (Hermy and Cornelis, 2000). Erosion prevention and water purification were the services least affected by park vegetation and topographic heterogeneity. Sediment retention and loss values were relatively low (data not shown) when compared to maximum values estimated for other larger river basins (e.g. Bangash et al., 2013; Hamel et al., 2015; Terrado et al., 2014). Vegetation reduces the production of sediments (Gómez-Baggethun and Barton, 2013; López-Vicente et al., 2013; Verstraeten et al., 2006), so the high vegetation cover in all vegetation types may be reducing the risk of erosion, even in the areas with more pronounced slopes (forest and grassland with high tree density). Nitrogen retention was high in all vegetation types, with the calculated amounts of retention and loss of nitrogen (data not shown) being generally low and close to the values estimated by Groffman et al. (2004) for forest areas, and lower than those obtained for suburban and agricultural areas by the same author. No differences were found between areas with low and high tree density, although the filtering capacity of vegetation may depend on its characteristics, such as the cover of trees or grass (Nidzgorski and Hobbie, 2016; Norris, 1993), with tree-only buffers potentially having a greater retention capacity. However, Nidzgorski and Hobbie (2016) also did not observe any differences in nitrogen leaching between urban turf and tree areas. It is important to say that both models have limitations, such as the sediment sources covered by the USLE equations used in the Sediment Delivery Ration model, or not considering chemical or biological interactions other than terrestrial vegetation in the case of the Nutrient Retention model (Sharp et al., 2016). The homogeneity of our results throughout the study area may result from these models' inherent limitations, but they are likely also the result of the park's planning. In fact, the park's terrain was modeled to help prevent floods at neighborhoods located downstream, so its construction was expected to greatly reduce water runoff, with consequences on both sediment and nutrient retention. Thus, despite the models' limitations, they may be suitable to help planners identify areas of potential intervention, even if a deeper assessment is needed to assist in their implementation. Finally, the air purification service, viewed here by lichens' richness, was only slightly higher in mixed forest, when compared to all other vegetation types. Other tools could give precise measurements on air quality and levels of pollutants (e.g. monitoring stations); however, they are expensive and are not economically feasible at this spatial scale. It is also important to note that one single assessment of lichen richness reflects the lichens' long-term response to pollution, which is considered a time-integrated measure of air pollution (Llop et al., 2017; Pinho et al., 2008, 2004; Santos et al., 2017). We expected vegetation types with higher tree density to present better air quality, because forests are efficient in trapping atmospheric particles (Kocic et al., 2014; Yang et al., 2005) and forest canopies can limit the mixing of upper level air with ground level air, which may lead to significant below-canopy air quality improvement (Nowak et al., 2006). However, our study area is likely subjected to a rather low pollutant concentration in general (Munzi et al., 2014). Thus, although some differences could be seen between the forest and the other areas, these differences were small. However, in other more polluted cities, the presence of trees can significantly ameliorate air pollution (Bodnaruk et al., 2017), and so trees, especially when planted in high densities, must be considered an important asset to the provision of air purification (Zupancic et al., 2015). In general, the metrics used to assess ES are widely applicable and cost-effective. Among these, lichen diversity requires the most skills, namely lichen species identification. However, the metrics used, lichen species richness and LDV of oligotrophic species, allowed us to assess the relative amount of two ES that are very complex to measure, namely air purification and habitat quality. Air purification can be measured using complex and expensive air monitoring stations, but that is not economically feasible at this spatial scale. Regarding habitat quality, its complete evaluation would require monitoring multiple biological taxa, which is also not possible. Seed

Fig. 3. Ecosystem services' relative supply according to vegetation type, showing scores attributed to each ecosystem service (values range from 0 to 1, with 1 being the maximum value measured in the park) and total mean score of each vegetation type (mean ± SD; n = 5; green – higher score, orange – lowest). Only vegetation types with all six evaluated ES are shown. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.).

dispersal also required sampling bird diversity, and followed a reasonable assumption that more seed-carrying birds can allow a higher possibility of seed dispersion. Bird sampling expertise is nowadays frequent, especially in urban areas, and thus this metric is rather cost-effective. Carbon sequestration was the most time-consuming task, especially field sampling of vegetation. This resulted in the most accurately estimated ES, and although it's timeconsuming, it can be performed by a non-specialist. Finally, water purification and erosion prevention were assessed using mostly GIS operation and bibliographic search. These services can be measured in the field by at an extreme economical cost for such a high number of sub-basins and orography and land-cover conditions. As such, the method that was employed in this study represents the most cost-effective way to quantify these services. Nevertheless, all metrics used would benefit from some field validation whenever possible. Although the impact of visitors was not assessed by our work, we assumed that this pressure did not significantly affect the evaluated ES. Vegetation characteristics did not seem to be modified by visitors, either because people preferentially use the lawns, which are more intensely managed, or because they keep to the trails when using other areas. Visitors' pressure has been shown to change soil and vegetation at a small scale (e.g. Sarah and Zhevelev, 2007), but high-pressure sites in our lawns are localized areas, so its impact is probably diluted at vegetation type scale. In fact, the service that we expected to be negatively impacted by visitors was seed dispersal, since birds can be sensitive to human presence (e.g. Fernández-Juricic and Tellería, 2000). Nonetheless, lawns had one of the highest bird counts in the park. Notwithstanding, future works should address visiting intensity to accurately determine its impact on ES. 4.2. Trade-offs and synergies in ecosystem services supply The assessment of multiple ES and the identification of trade-offs and synergies among them allow the manipulation of ecosystems to optimize their provision and promote their sustainability (Bennett et al., 2009; Rodríguez et al., 2006). Urban ecosystems may only provide part of the ES that cities require, but this provision may be significant in highly populated areas (Gómez-Baggethun and Barton, 2013). Moreover, a multifunctional perspective on planning and management can bring more value to green spaces than focusing on a few ES, by promoting an agreement between community members, investors and decision-makers (Lovell and Taylor, 2013). Multifunctional green spaces seek to provide different ES, balancing the needs of local residents and the society as a whole (Lovell and Taylor, 2013). Our study area has been managed towards users' needs and was designed to fulfil mostly recreational and aesthetical services, although its construction took other services into account (e.g. water drainage). In the past, urban green spaces were often designed as homogeneous ecosystems, but in larger parks such as this one (area of 44 ha), the heterogeneity in vegetation types creates different habitats that will consequently produce different ES provision settings. At a larger extent, city-scale assessments may allow planners to balance ES provision through different green spaces, including smaller urban parks with only one land-cover type, or other areas, such as street trees. Finally, ecosystem disservices in urban areas (i.e. ecosystem functions that are perceived as negative for human well-being; Lyytimäki and Sipilä, 2009) should also be considered in trade-off assessment, since they can have an important weight on the functioning and value of green spaces. Regarding the ES assessed in this study, their optimization may be accomplished by improving forests' characteristics in built areas. For example, by increasing tree density, areas with lower score on carbon sequestration will gain value and air purification can also be improved. Furthermore, if vegetation handling is less intense and understory structure is incremented, habitat quality may also get better. On the other hand, seed dispersal service may decrease, which may be counterbalanced by the maintenance of open areas and lawns. Although the vegetable garden was only partially evaluated, this type of land-cover has the potential to provide several important ES, in some cases even more than urban parks (Speak et al., 2015), so its integration could also be a good management decision that would improve a park's overall functioning. 4.3. Conclusions In this work, we provide a general framework to optimize the management of urban parks as multifunctional green spaces, through the spatially detailed assessment of ecosystem services. By using this approach, we can identify trade-offs and synergies in the provision of ecosystem services, providing important information for both the management and planning of urban parks. The assessment at park level can give local authorities an extra tool to manage areas based on landcover heterogeneities. We focused on vegetation handling, since this is a common practice in most urban parks. Thus, ecosystem services that are more easily related to vegetation type are a good starting point for its inclusion in management plans, since ecosystem services' provision can be adjusted with the implementation of simple measures. The simultaneous assessment of various ecosystem services requires multidisciplinary teams, whose costs may be pointed out as the main limitation for future applications of this approach. However, there is an increasing number of parks that have information on biodiversity that could help ecosystem services' evaluation by non-specialists. For example, some parks perform birds census, which could help to address the seed dispersal service, or map trees, which would save time in the estimation of carbon stocks. Also, results from one park's assessment could be applied to nearby parks' planning or management. This would allow a broader implementation of management actions for ecosystem services' promotion, while reducing associated costs. Despite these limitations, we showed that urban parks' management and planning can make use of simple practical measures as naturebased solutions to promote ecosystem functioning in urban areas and to manage green spaces for the provision of specific ecosystem services. Finally, these solutions should be monitored and their impact on ecosystem services assessed, in order to identify further trade-offs and winwin situations and improve the success of green infrastructures in urban areas.

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Appendix A. Supporting information Supplementary data associated with this article can be found in the online version at http://dx.doi.org/10.1016/j.envres.2017.10.023.

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